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A comparative study of nitrate leaching from intensively managed monoculture grass and grass–clover pastures

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SUMMARY

Nitrate leaching losses from intensively managed monoculture grass and grass–clover pastures were measured during 1994–96 at a long-term experimental farm in south-west Scotland. Field-size lysimeter plots were established in 1993 on the existing pastures on a silty clay loam non-calcareous gley. No fertilizer-N was applied to the grass–clover, while the monoculture grass was fertilized with *c*. 240 kg N ha^{-1} $year^{-1}$, but both swards received 2–3 cattle slurry applications annually (120–390 kg total N ha^{-1} $year^{-1}$). The pastures supported 2–3 cuts for silage conservation, and were grazed by dairy cattle and stocked with sheep during the winter months.

Initially, leachate nitrate concentrations from the fertilized grass were considerably larger than those from the clover-based pasture, but became similar with time. The annual nitrate leaching losses from the grass–clover (24–38 kg NO_3-N ha^{-1}) were less than that from the monoculture grass (30–45 kg NO_3-N ha^{-1}), but the differences were not large considering the additional fertilizer-N applied to the latter treatment. Results also suggested that greater leaching losses occur during a warmer, drier year, compared to a cooler, wetter year, regardless of the source of N-input.

INTRODUCTION

Increased leaching of nitrate-N from agricultural soils is a major environmental concern. Consequently, agricultural practices which influence nitrate leaching have been the focus of considerable scientific interest (The Royal Society 1983; Ryden *et al*. 1984; Germon 1989; Addiscott *et al*. 1991). In recent years, research findings have presented sufficient evidence that nitrate leaching from sites with similar soils and environmental conditions increases with increasing fertilizer-N applied (Barraclough *et al*. 1983; Watson *et al*. 1992; Scholefield *et al*. 1993; Jemison *et al*. 1994), and that greater losses occur from soils under arable crops than those under perennial grass production systems (The Royal Society 1983; Burt & Arkell 1987).

Nitrate leaching from swards that are cut rather than grazed is considered to be relatively small (Dowdell & Webster 1980; Barraclough *et al*. 1983), but losses from intensively fertilized (400 kg N ha^{-1} $year^{-1}$) grazed pastures have frequently exceeded 150 kg NO_3-N ha^{-1} $year^{-1}$ (Ryden *et al*. 1984; Scholefield *et al*. 1993). These large differences between cut and grazed pastures are considered to be largely due to the recycling of N in the excreta of grazing animals (Cuttle & Jarvis 1992). While these studies with cut or grazed grass production systems have provided useful information on the influence that grazing activities may have on nitrate leaching, the results may not be representative of most field situations where pastures receive regular slurry applications, and are grazed as well as cut for silage conservation. Clearly, nitrate leaching losses from grassland farms in their totality need to be quantified, rather than on the basis of the practice in particular fields (e.g. grazing or cutting).

Lowland grassland farms in the UK and other countries of north-western Europe are mainly systems with intensive fertilizer-N inputs, and excessive nitrate leaching from such units appears to have prompted interest in reduced fertilizer-N grass production systems which rely mainly on biological fixation of atmospheric-N by legumes such as clover (*Trifolium* spp.) and/or recycling of slurry/manure-N. There is some evidence that nitrate leaching losses from

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grass–clover pastures are much smaller than from pastures fertilized with mineral-N (Ryden *et al.* 1984; Parsons *et al.* 1991; Ruz-Jerez *et al.* 1995). However, much of this evidence is based on intensively fertilized (400 kg N ha^{-1} $year^{-1}$) pastures. In contrast, leaching losses from a clover-based pasture (35% clover content) were larger than from moderately fertilized (150–200 kg N ha^{-1} $year^{-1}$) monoculture grass, although the trend was reversed when the clover content of the mixed pasture decreased significantly (Cuttle *et al.* 1992). Since there have been few measurements of nitrate leaching from grass–clover pastures, it is commonly perceived that clover-based grass production systems are environmentally benign. More information is therefore needed for quantitative assessment of these so called 'environmentally friendly' grass production systems.

In this paper, we report the results of nitrate leaching from two contrasting grass production systems: monoculture grass and grass–clover swards. The main objective of this study was to compare nitrate-N leaching from the two grass production systems, both of which were intensively managed (cut, grazed and slurry-applied), but differed in fertilizer-N input.

MATERIALS AND METHODS

Experimental site and soil

The study was carried out from 1994 to 1996 at the Acrehead study unit of the Crichton Royal Farm (NX 978727), Dumfries, south-west Scotland. The average annual precipitation at the farm is 1054 mm (64-year period, 1931–94). The soil at the experimental site is a non-calcareous gley with a silty clay loam topsoil over a silty clay subsoil. Typical properties of the soil are given in Table 1. Like the experimental site, the soils in this area are mainly non-calcareous gley of the Stirling Association (FAO – Eutric gleysol), and were developed on estuarine and lacustrine raised beach silts and clays at *c.* 15 m above sea level. Long-term meteorological records indicate that soils in this area usually return to field capacity about mid-October and cease draining by early April, and that most soil drainage is likely to occur between November and March. During the present study, the drainage season extended from October of one year through to April of the following year. Daily precipitation and soil and air temperatures were obtained from Meteorological Office Station No. 6641 (Crichton Royal, Dumfries) which is *c.* 1 km from the experimental site. Precipitation and temperature data for the study, and long-term averages, are presented in Table 2.

Grass production systems

Since 1977 the Acrehead unit has been under monoculture perennial ryegrass, receiving *c.* 350 kg fertilizer-N ha^{-1} $year^{-1}$. Previously it was under a rotational arable cultivation system. For the purpose of comparing productivity and profitability of a clover-based dairy herd with a high fertilizer-N (350 kg N ha^{-1}) system, the two 36 ha units, monoculture grass and grass–clover, were established by ploughing and reseeding the existing pastures during July 1987 and April 1988 (J. A. Bax & G. E. D. Tiley, unpublished). The grass–clover unit was reseeded with ryegrass (*Lolium perenne* cvs Merlinda and Morgana) and white clover (*Trifolium repens* cvs Milkanova and Menna), and the monoculture grass unit contained only ryegrass. Since 1991 fertilizer-N input to the monoculture grass has been reduced to *c.* 250 kg N ha^{-1}. The fertilizer is applied in three top-dressings generally between mid-March and mid- to late July. While urea is used for the March application, the latter two dressings are made with ammonium nitrate. These swards are cut 2–3 times per year for silage, as well as being grazed by dairy cattle at a stocking rate of 2 cows ha^{-1}, and stocked with sheep during the winter months (December–March), when the cattle are housed. Sheep grazing on the two

Table 1. *Some physical and chemical properties of the soils (sampled in April 1994) at the experimental site in Dumfries, SW Scotland. Figures in parentheses are for the 40–60 cm soil depth*

	Grass	Grass–clover
Particle size distribution (0–20 cm) %		
Clay (< 0·002 mm)	25·3 (44·6)	26·6 (42·8)
Silt (0·002–0·063 mm)	55·9 (49·4)	58·7 (53·2)
Sand (0·063–2 mm)	18·8 (6·0)	14·7 (4·0)
Dry bulk density, kg m^{-3} (0–20 cm)	1·33	1·33
Loss-on-ignition (%)*	8·10	8·15
Oxidizable organic C (%)*	3·46	3·41
Total N (%)*	0·29	0·29
pH (1:2·5 w/v, in water)*	5·44	6·16

* For the 0–6 cm soil.

Table 2. *Summary of precipitation and temperature records obtained from Meteorological Office Station No. 6641* (*Crichton Royal, Dumfries*)

	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Annual
Precipitation (mm)													
64-year mean	58	67	65	83	89	100	113	111	113	110	71	74	1054
1993/94	140	100	52	60	43	53	47	68	216	161	95	116	1151
1994/95	68	15	54	105	91	52	86	138	172	142	129	102	1153
1995/96	20	79	19	63	17	71	164	85	39	91	225	96	967
Air temperature (°C)*													
64-year mean	7·4	10·3	13·2	14·8	14·5	12·3	9·4	5·9	4·2	3·2	3·5	5·2	8·6
1993/94	8·3	9·9	13·3	13·4	12·9	11·0	7·3	3·9	3·4	3·7	2·4	5·6	7·9
1994/95	6·9	9·3	11·3	15·1	13·5	10·9	8·8	8·7	5·4	3·6	3·6	3·7	8·4
1995/96	7·3	10·2	13·7	16·3	16·8	12·1	12·0	7·3	1·5	4·8	3·4	4·2	9·1
Soil temperature (°C)†													
1993/94	8·4	10·8	13·8	14·9	14·4	12·9	9·5	6·3	4·6	3·7	3·1	5·3	8·9
1994/95	7·6	10·5	12·4	15·7	15·9	13·3	10·1	9·7	6·7	4·4	5·1	4·9	9·7
1995/96	7·8	11·2	13·7	16·2	17·1	14·5	12·9	9·4	5·9	4·9	3·8	5·3	10·2

* Average daily mean air temperature; † Monthly mean soil temperature at 30 cm depth.

Table 3. *Summary of fertilizer- and slurry-N inputs* (*kg N ha*$^{-1}$); *cattle slurry applications were at the rate of 50 m*^{3} *ha*$^{-1}$, *there were two in 1994/95* (*May and November*) *and three in 1995/96* (*May, July and January*)

Year	Grass	Grass–clover
1994/95		
Fertilizer	245	0
Slurry-N		
NH_4-N	80	65
Organic-N	51	56
Total slurry-N	131	121
Total (fertilizer + slurry) N input	376	121
1995/96		
Fertilizer-N	235	0
Slurry-N		
NH_4-N	172	170
Organic-N	147	220
Total slurry-N	319	390
Total (fertilizer + slurry) N input	554	390

pastures is regulated on a monthly rotation basis, with a stocking density of 44 lambs ha^{-1}.

The grass–clover sward does not receive any fertilizer-N; however, both pastures receive 2–3 slurry applications annually. Although both units receive an equal number of slurry applications, slurry-N inputs can be different because each unit has separate housing, silage pits and slurry storage. A record of fertilizer- and slurry-N inputs to the pastures during the 1994/95 and 1995/96 leaching seasons is presented in Table 3.

Drainage measurement and sampling

The drainage monitoring facility at the farm comprised four field-size lysimeter plots, each with an area of *c.* 0·5 ha, and was established in the summer of 1993. Two of the plots were on the monoculture grass field and two on the grass–clover field. These fields are on either side of a farm road, and are integral parts of the two grass production systems described in the preceding section. The plots were isolated from one another, and from the area outside the study site, by using 'curtain' and 'interceptor' drains respectively. This, together with almost flat topography and relatively high hydraulic conductivity values of the surface layers (McGechan *et al.* 1997), makes each plot hydrologically isolated, except for deep percolation. However, because of much reduced hydraulic conductivity values of the subsoil layers (0·50–1·0 m) and the drains being laid on an impermeable layer, deep percolation from the plots was likely to be low (McGechan *et al.* 1997).

Each lysimeter was drained by a main drain (PVC, perforated pipe laid at a depth of 0·9 m) running along the down-slope periphery of the experimental site. The main drain was fed by lateral drains with 7 m spacing and laid at a depth of 0·7 m. Outflows from each plot flowed through separate V-notch weirs. The drainage measurements started in February 1994, and initially drainflow from each plot was measured using the mechanical head recording system described by Talman (1983). This system, although reliable and accurate, was not compatible with the automated drainage sampling equipment because flow- or volume-proportional sampling was not feasible. Consequently, at the beginning of the 1994/95 leaching season, a non-contacting (ultrasonic probes) electronic flow metering device (Detec 2020, Montec International Ltd, Manchester) was installed on the existing mechanical head recording equipment. Weekly drainflow data were retrieved using a portable computer. The drainage measurements made by the electronic device were found to be similar to those estimated by the mechanically recorded hydrograph (Talman 1983).

Volume-proportional drainage samples were collected using automated samplers (Epic 1011, Montec International Ltd, Manchester). The sampling regime was varied depending on the antecedent and anticipated weather and soil conditions. During the winter drainage period, the water sampling frequency generally varied between 0·5 and 2 mm drainage. The samplers were programmed to collect samples for every 0·05 mm or less drainage during the summer months. Individual samples from the sampling equipment were collected in 500-cm^3 capacity plastic bottles during a weekly visit to the site. Water containers in the sampling equipment were cleaned of any sediments and, if required, 0·5 cm^3 dilute sulphuric acid (H_2SO_4) was added to each container (1-dm^{-3} capacity) before re-programming the equipment for the next week. The use of H_2SO_4 prevented biodegradation of samples.

Water and soil analysis

Where > 12 water samples were obtained from a given lysimeter, the samples were bulked in such a way that no more than 10 samples per week per lysimeter were processed for chemical analysis. The water samples were filtered through 0·45 μm membrane filters (Millipore HVLP) on the same day before being refrigerated (< 5 °C). The filtered samples were analysed for nitrate-N within 24 h of their collection, using a colorimetric method (Jackson 1958). During 1995/96, the pastures were also sampled for soil mineral-N (NO_3-N and NH_4-N), determined following extraction in 2 M KCl (Rowell 1994). Soil pH, loss-on-ignition and oxidizable organic carbon were analysed using standard procedures as described by Rowell (1994). Total N (Kjeldahl N) in soil and slurry and mineral-N in slurry were analysed using methods described by Greenberg *et al.* (1992).

Assessment of nitrate loss in drainflow

An average weekly nitrate concentration was calculated for individual plots. These weekly nitrate concentrations on replicate plots for both monoculture grass and grass–clover were very similar. Nitrate loss in drainflow was calculated as the product of weekly mean nitrate concentration and the total measured drainage for the week. The four plots (two on each system) had different drainage yields due to variations in drain efficiency. The drainage yield of one of the plots that closely matched the expected drainage output from the site (McGechan *et al.* 1997) was therefore used for the purpose of computing weekly and hence annual nitrate leaching losses. Annual nitrate leaching losses include the total loss over the entire winter period and any output that occurred during the preceding summer and autumn months.

RESULTS

The year 1994/95 had more than the long-term average precipitation recorded at the nearby Meterological Office Station (Table 2); in contrast, 1995/96 was drier than average, and had a summer season considerably warmer than those of 1994/95 and the long-term average (Table 2). In 1994/95, the rainfall was evenly distributed over the winter months (Fig. 1), consequently the soil moisture content remained at or above field capacity over almost the entire drainage season with a total measured drainage of 782 mm. During 1995/96 there was a contrasting pattern of rainfall, particularly over the winter months, with prolonged dry spells often followed by short-duration large drainage events. Of the total 440 mm drainage recorded during 1995/96, one single snowfall event in February 1996 alone accounted for > 150 mm of drainage (Fig. 1). Estimates were made of expected drainage yields during the two contrasting periods 1994/95 and 1995/96, using the BUCKET model (Rowell 1994). The results for 1994/95 indicated that the soil water deficit (SWD) was zero from mid-October until the first week of April 1995, with a

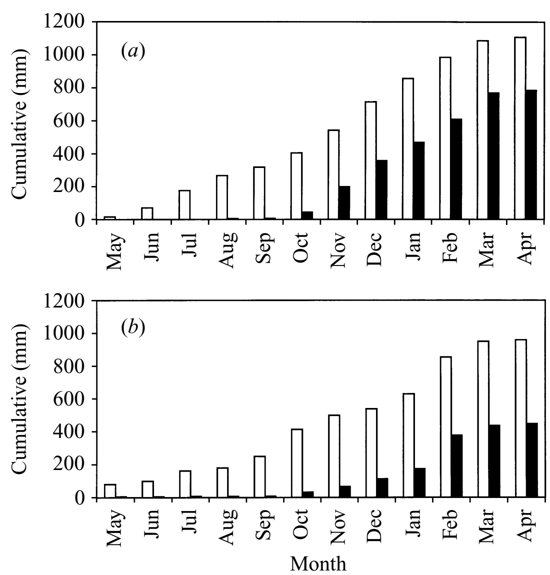

Fig. 1. Rainfall (□) inputs (mm) and drainage (■) outputs (mm) during the (*a*) 1994/95 and (*b*) 1995/96 leaching seasons.

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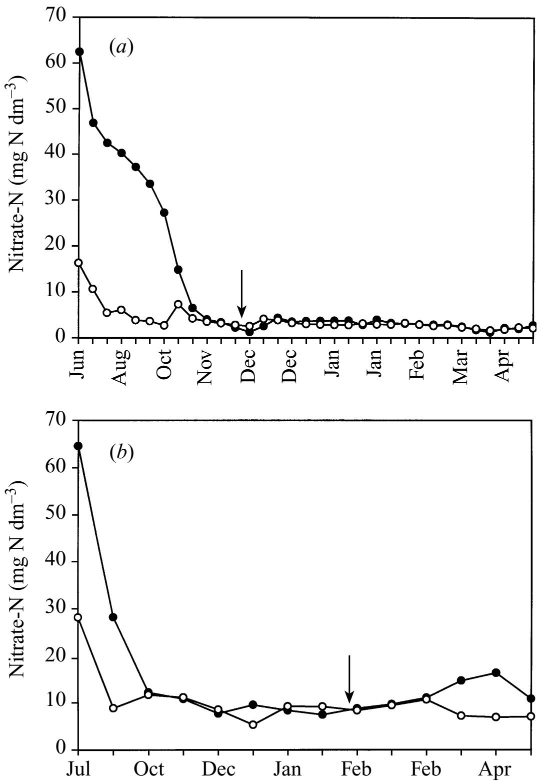

Fig. 2. Chemographs of weekly-average nitrate-N concentrations (mg dm^{-3}) in drainflow from the grass (●) and grass–clover (○) pastures during (*a*) 1994/95 and (*b*) 1995/96. Both swards received cattle slurry applications but fertilizer-N was applied only to the monoculture sward. The arrows show when slurry was applied during brief dry periods.

maximum SWD of 97 mm in June 1994. During 1995/96, the deficit was zero from the beginning of October 1995 to the end of March 1996, except for the first week of 1996, which had a deficit of 13 mm, and the maximum (129 mm) was in mid-August 1995. The different input patterns for the two leaching seasons are obviously largely responsible for the large differences seen in the drainage yields for 1994/95 (68 %) and for 1995/96 (46 %).

Drainflow measurement and water sampling at the site began in February 1994. Nitrate concentration data (< 1–5 mg NO_3-N dm^{-3}) for the samples collected between February and April 1994 indicated no major differences between the two grassland systems. However, since we were not able to monitor the drainage from the beginning of the 1993/94 leaching season, the results primarily of the subsequent two complete years are discussed here, i.e. 1994/95 and 1995/96.

Figure 2 presents the chemographs of nitrate concentrations in drainflow from the two grass production systems. During the 1994/95 leaching season, weekly-average concentrations of nitrate ranged from 1·1 to 62·5 mg NO_3-N dm^{-3} for the fertilized monoculture grass, and from 1·5 to 16·3 mg NO_3-N dm^{-3} for the grass–clover (Fig. 2*a*). The highest nitrate concentrations in the drainage water were measured after heavy rainfall events during summer months in both years, and that is when the largest differences between the two swards occurred (Fig. 2). Nitrate concentrations in the drainage water from both swards decreased as the winter drainage volume increased. After some initial leaching events, weekly-average nitrate concentrations in the water from the fertilized grass and the grass-clover that received no fertilizer-N were similar, and remained consistently < 4 mg NO_3-N dm^{-3} for the remainder of the leaching season (Fig. 2*a*).

The maximum weekly-average nitrate concentrations from the grass plots were similar during both years, but that from the grass–clover plots (28·1 mg NO_3-N dm^{-3}) during the second year was much larger (Fig. 2). As in 1994/95, during 1995/96 also, the nitrate concentrations from the two pastures decreased to comparable contents as leaching progressed. However, the concentrations remained larger than in the previous year (Fig. 2). The nitrate chemographs clearly show that during both years the concentrations of nitrate in drainage water from the grass and grass–clover pastures differed only during the early leaching season (Fig. 2). This is consistent with the leaching trend observed during the February–April 1994 period (1993/94 leaching season) when drainage monitoring began for the first time.

Slurry application during wet winter months is not recommended; however, the practice is common among dairy farmers faced with limited slurry storage capacity. As in any other commercial farm, cattle slurry at the rate of 50 m^3 ha^{-1} was applied to both pastures during a brief dry period in late November 1994 (1994/95 leaching season). Slurry effluents were seen in the drainflow in the following week after a rainfall event, the water being dark brown at the beginning and becoming progressively lighter in colour. This slurry application, however, had no impact on nitrate concentrations in the drainage water (Fig. 2*a*). A similar slurry application in January 1996 when the soil was frozen, which was followed by > 150 mm of snowfall, had no effect on the concentration of nitrate in the subsequent snowmelt-induced drainage (Fig. 2*b*). Nitrogen in stored cattle slurry occurs essentially in ammonium and organic forms (Evans *et al.* 1980). It was therefore not an unexpected observation that the slurry applications when the soil was close to saturation or was frozen had no effect on the concentration of nitrate in the subsequent drainage. Since slurry effluents were visible

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Table 4. *Nitrate concentrations (mg NO_3-N dm^{-3}) and outputs (kg NO_3-N ha^{-1}) for fertilized grass and grass–clover pastures; the concentration ranges are based on weekly-average concentrations*

	1994/95			1995/96		
	Grass		Grass–clover	Grass		Grass–clover
Mean concentration*	3·9		3·1	10·2		8·5
Concentration range	1·1–62·5		1·5–16·3	7·6–64·6		5·2–28·1
Nitrate leaching output	30·2		24·3	44·8		37·7
S.E.D.		3·27			4·51‡	
D.F.†		32			12	

* Based on flow-weighted concentration data.
† Degrees of freedom (n-1) where n is weekly-average nitrate concentration/output.
‡ The difference in nitrate leaching outputs being non-significant ($P > 0·05$).

in the drainage water, it should be recognized that these winter slurry applications must have had contributed slurry-borne organic- and NH_4-N to the water. Drainage from slurry applied fields, as in the present study, and farmyards can cause a considerable deterioration in water quality. Schofield *et al.* (1990) found that farm streams receiving such drainage had high BOD (biochemical oxygen demand) and ammonia concentrations.

The nitrate concentration and output data for the 1994/95 and 1995/96 seasons are summarized in Table 4. Although during both years flow-weighted mean nitrate concentrations for the grass–clover (3·1–8·5 mg NO_3-N dm^{-3}) were smaller than those for the monoculture grass (3·9–10·2 mg NO_3-N dm^{-3}), the differences were not large (Table 4). A striking feature of the data was that the annual nitrate concentrations for both swards increased by > 2·5 times during the second year compared to the first. During the 1994/95 leaching season, nitrate-N leaching losses from the grass–clover (24·3 kg N ha^{-1}) were equivalent to 80% of that lost from the fertilized grass (30·2 kg N ha^{-1}). Statistical analysis of the data using a *t*-test on pairs of weekly nitrate-N quantities leached showed that this difference in leaching losses between the two grass production systems was significant ($P = 0·03$).

The difference in nitrate-N leaching losses between the monoculture grass (44·8 kg N ha^{-1}) and the grass–clover (37·7 kg N ha^{-1}) during 1995/96 was relatively small compared to the previous year, and this difference was not statistically significant ($P > 0·05$). The results also showed that nitrate leaching losses were considerably larger during 1995/96 than during 1994/95 regardless of the grass production system (Table 4).

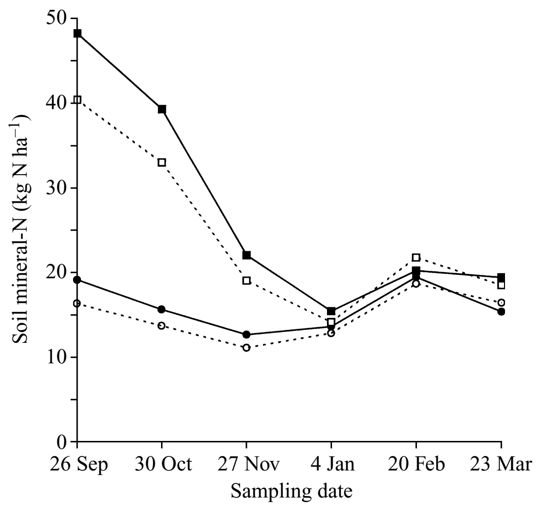

Fig. 3. Soil mineral-N (kg ha^{-1}) measured during 1995/96 as nitrate-N (●) and ammonium-N (■) in the grass field, and nitrate-N (○) and ammonium-N (□) in the grass–clover field. The values are for the 0–30 cm soil layer and are expressed on an oven-dry mass basis. Cattle slurry was applied on 31 January 1996 to both pastures. Obviously, soil mineral-N may have been even larger than seen in the February sampling.

DISCUSSION

Nitrate leaching pattern

The leaching losses of nitrate-N from the monoculture grass (Table 4) are similar to those found in other similarly conducted studies of comparable fertilizer-N input pastures in the UK (Watson *et al.* 1992). Nitrate-N leaching losses from a pasture similar to that of the present study (i.e. cut, grazed and slurry-applied) varied between 17 and 35 kg N ha^{-1} over a 2-year period (Jordon & Smith 1985). Over a 9-year period, Scholefield *et al.* (1993) found variable annual nitrate-N leaching losses of between 20–54 and 66–186 kg N ha^{-1}, respectively, from continuously cattle-grazed grass plots with annual fertilizer-N inputs of 200 and 400 kg ha^{-1}. These rather large year-to-year differences in nitrate leaching were attributed to the pattern of summer weather, and it

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was concluded that twice as much nitrate was leached after a hot dry summer than after a cool wet one. In the present study, the cumulative drainage during 1995/96 (440 mm) was much less than that in 1994/95 (782 mm), but the leaching losses from both pastures were larger. The elevated nitrate leaching during 1995/96 might have been due in part to the relatively warmer weather during the months of May–October 1995 (Table 2), and to a pattern of winter rainfall which was punctuated by prolonged dry spells. This reaffirms the earlier finding that there is a strong influence of antecedent climatic conditions, with dry warm summers being followed by intense leaching in subsequent rainfall events (Jordon & Smith 1985; Burt *et al*. 1988; Trudgill *et al*. 1991; Scholefield *et al*. 1993).

The effects of weather on nitrate leaching are not simple since the former (e.g. temperature, rainfall) affects the N-cycle by influencing mineralization, nitrification, denitrification and soil aerobicity (Jarvis *et al*. 1996). A warm summer as during 1995 in the present study (Table 2), is therefore likely to result in enhanced mineralization and potentially greater nitrate leaching losses compared to a cool summer. Furthermore, prolonged dry spells between rainfall events as observed during 1995/96 may also have contributed towards increased nitrification and consequently greater leaching in the subsequent wet periods due to a 'drying-rewetting' effect (Birch 1960). In addition to the favourable antecedent climatic conditions, much larger slurry-N inputs during 1995/96 compared with those in 1994/95 (Table 3) would also be a contributory factor to the increased leaching during 1995/96, a conclusion further supported by soil mineral-N data which showed a rise in both NH_4-N and NO_3-N following slurry application in January 1966 (Fig. 3).

During 1995/96, soil mineral-N was also monitored and the data showed that soil nitrate-N supply did decrease during the early leaching season (Oct–Nov), probably due both to continued plant uptake and to leaching losses (Fig. 3). After this initial period, the soil-nitrate supply did not show any major reduction (Fig. 3). The slurry application in January 1996 which contributed the majority of the total slurry-N input, together with favourable conditions for mineralization/nitrification, appears to have maintained a relatively stable soil-nitrate supply during 1995/96. Such a relatively large and stable soil-nitrate supply is consistent with nitrate concentration in the drainflow (Fig. 2*b*), and clearly demonstrates that nitrate leaching from the pastures was transport-limited (i.e. more rainfall during the 1995/96 winter months would have led to even greater leaching losses). The small nitrate concentrations in the 1994/95 drainflow (Fig. 2*a*) would suggest that during that year, nitrate leaching had reached a supply-limited situation. These observations support the earlier hypothesis that supply-limited situations occur in cooler, wetter years and transport-limited situations in warmer, drier years (Burt *et al*. 1988).

Differences in nitrate leaching between grass and grass–clover

The leaching losses of nitrate-N from the grass–clover were equivalent to 80–84% of that lost from the monoculture grass fertilized with 240 kg N ha^{-1}. Although leaching from the grass–clover during 1994/95 was significantly smaller than that from the fertilized grass, the difference was not large. This lack of large differences in nitrate leaching between the two grass production systems suggests a similarity in total N inputs. Atmospheric-N fixation by clover in the grass–clover pasture was not assessed in the present study. However, white clover in grass–clover pastures in the UK has the potential to fix 100–200 kg N ha^{-1} $year^{-1}$, depending on the proportion of clover (Wood 1996). The clover content in the grass–clover pasture averaged 20 and 27% (w/w) during 1994 and 1995, respectively. A figure of *c*. 100 kg biologically fixed N ha^{-1} $year^{-1}$ would probably be a reasonable estimate for this pasture (Kristensen *et al*. 1995), and would bring the total N input in the grass–clover to a level that is not very different from that of the fertilized grass (Table 3). During 1995/96, nitrate leaching from the grass–clover was 16% less than that from the fertilized grass but the difference was not statistically significant. Relative to the grass, the 1995/96 input of slurry-N in the grass–clover pasture was much larger compared to 1994/95 (Table 3) and this, together with the increase in clover content from 20 to 27%, may have been contributory factors to relatively larger leaching from the grass–clover, narrowing the leaching loss difference between the two pastures. In addition, soil mineral-N data also suggested that the leaching losses from the two pastures would be similar (Fig. 3).

This comparison has shown that nitrate leaching from an intensively managed, i.e. cut, grazed and slurry-applied, grass–clover pasture is likely to be at least similar if not lower than that from a similarly managed and moderately fertilized (240 kg N ha^{-1}) grass pasture. The comparison also indicated that nitrate leaching from a clover-based grass production system such as that in the present study is likely to be much less than that from the intensively fertilized pastures of other recent studies (Parsons *et al*. 1991; Ruz-Jerez *et al*. 1995). The environmentally benign nature of legume-based pastures was demonstrated in a recent Ohio study where beef cattle grazed orchard grass (*Dactylis glomerata* L.) and tall fescue (*Festuca arundinacea* Schreb.) pastures which were fertilized with 224 kg N ha^{-1} $year^{-1}$ for 5 years. At the beginning of the sixth year, lucerne (*Medicago sativa* L.) was interseeded into the pastures and fertilizer application discontinued (Owens *et al*. 1994). Nitrate-N concentrations in groundwater collected from purpose-built springs dropped, on average, by 150% during a 2-year period before further decreasing to the pre-fertilization levels.

In contrast, Cuttle *et al*. (1992), using ceramic suction cup samplers, estimated greater nitrate leaching from a clover-based pasture compared to a grass pasture fertilized with 150–200 kg N ha^{-1}. However, these authors reported large variations in nitrate leaching from the two pastures over 3 years, variations which were inconsistent with fertilizer-N input and summer weather, and which could only be partly explained. In view of the relatively short duration of the present study and the contradictory findings of Cuttle *et al*. (1992), further work is required to better assess the effects of fertilizer-N input and sward clover content on nitrate leaching.

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